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The response of phytoplankton to increased temperature in the Loviisa archipelago, Gulf of Finland

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Phytoplankton was studied in the Loviisa archipelago (south coast of Finland) in 1971–1994. Since 1977, thermal effluents from the Loviisa nuclear power plant have been discharged into the sea there. A general increase of nutrients in the Gulf of Finland contributed to an increase in phytoplankton biomass and primary production in the Loviisa archipelago during the late 1970s and 1980s, though biomass seemed to decrease again in the 1990s. The rise in temperature was, however, a more important factor than nutrients stimulating the production and biomass in the area close to the cooling water outlet. The thermal discharges increased especially the biomass of *Aphanizomenon* spp., which is the most abundant cyanoprocaryote (blue-green alga) and one of the most common phytoplankton taxa in the study area. At the intake area, total amounts of phosphorus best explained the changes in total biomass. The results indicate that increased temperature can lengthen the growing season, advance eutrophication and somewhat change species dominances in the circumstances prevailing in the northern Baltic Sea.

Introduction

During the last decades, thermal discharges into the aquatic environment from electricity industry have been brought into the public eye as an ecological concern (Langford 1990). Especially, in the conditions specific for the northern Baltic Sea, where the biota is adapted to seasonal variation with a cold icewinter and a temperate summer, the increase of temperature may cause increased environmental stress to the organisms. Furthermore, owing to the brackish water character of the Baltic Sea, many organisms exist near the limit of their physiological tolerance and have poor resistance to additional stresses (Dybern and Fonselius 1981). The effects of heated effluents are therefore of particular interest here. Thermal discharges may increase growth rates of primary producers, but the distinction of the thermal effects from those caused by the increase of nutrients poses a challenge especially in the Gulf of Finland, where the levels of phosphorus and nitrogen have significantly increased during the last decades.

The nuclear power plant at Loviisa came into operation in 1977. Extensive environmental studies were already started in the Loviisa archipelago in 1966 and they are still going on in the sea area surrounding the site of the power plant (Fig. 1). Thus, plenty of data are available

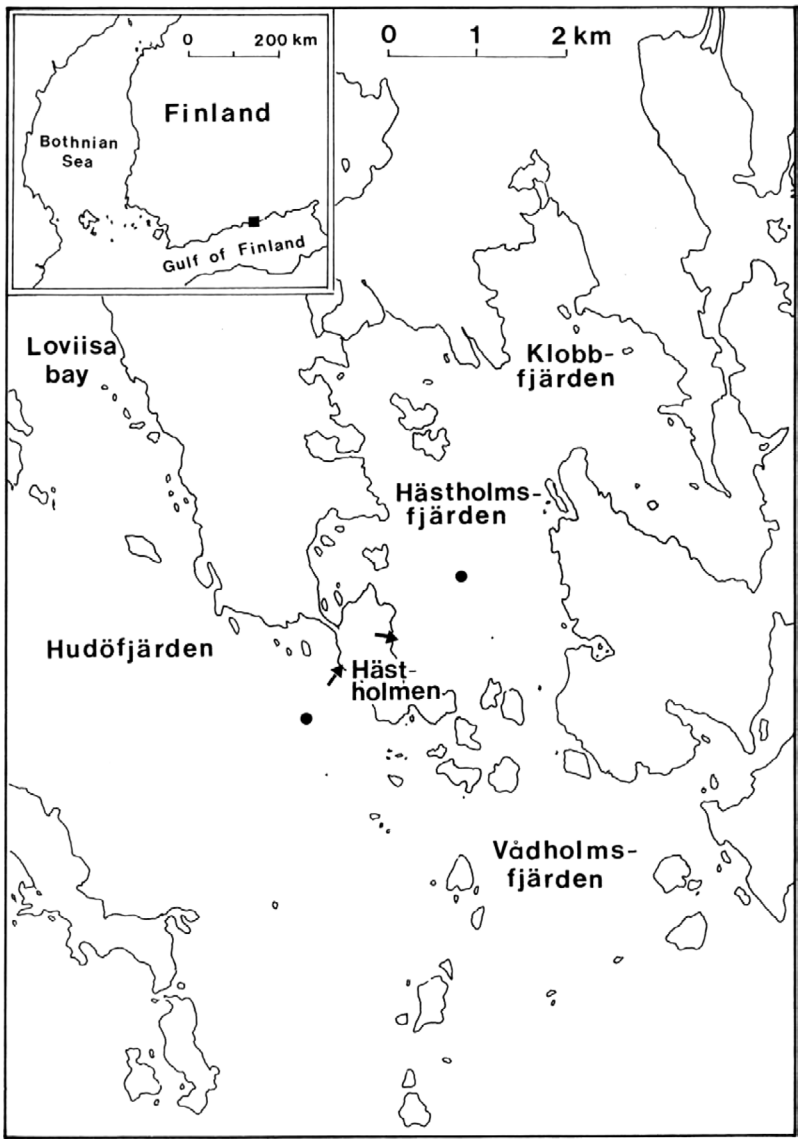

Fig. 1. Phytoplankton sampling stations in the Loviisa archipelago. The cooling water intake and outlet of the nuclear power plant are marked with arrows.

from the Loviisa archipelago (both before and after the power plant became operational) for evaluating the thermal effects of cooling water discharged into the sea from the power plant. Bagge and Voipio (1967) and Bagge and Niemi (1971) studied hydrography, the conditions on the seabed and the dynamics of phytoplankton in the late 1960s. Since then, studies on hydrography, phytoplankton and its primary production, zoobenthos and littoral vegetation have been continued with intensified permanent monitoring programmes, and we have summarized earlier data on phytoplankton biomass, species composition and aquatic macrophytes in the area for the years 1971–1982 (Ilus and Keskitalo 1980, 1986, 1987).

Brackish sea water is used for cooling in Finnish nuclear power plants. When passing through the cooling system, phytoplankton undergoes a sudden temperature rise. The impact of this temperature rise on algae cells can be either beneficial or harmful, depending on nutrient conditions, the physiological stage of the plankton community and the species composition (Keskitalo 1987a). The discharged cooling water transfers heat energy to the receiving water

body where phytoplankton is thus exposed to higher temperatures than in neighbouring sea areas.

In general, long-term laboratory cultures were a common way to study the effect of temperature on phytoplankton in the 1960s–1970s. For instance according to Goldman (1977a, 1977b), the response of phytoplankton to different temperatures is strongly species-dependent. However, the circumstances in nature are much more variable than in laboratory cultures. Our aim was to study the effect of temperature rise on phytoplankton production, biomass and species composition in sea areas receiving thermal effluents (in this case Loviisa archipelago), which can be interpreted as large-scale natural experimental areas. In addition, the aim of this study was to provide additional data to our earlier phytoplankton biomass and species composition results and to summarize the present data on the influence of temperature increase on phytoplankton and its primary production in the Loviisa archipelago. As we have also studied phytoplankton in the sea area near the Olkiluoto nuclear power plant on the west coast of Finland in 1972–1982 (Keskitalo 1987a, 1987b), differences in the responses of phytoplankton to increased temperature in these two sea areas are discussed. In addition, the effects of increased temperature due to the possible climatic change (Carter *et al*. 1995) in the coastal areas of the northern Baltic Sea are considered.

Material and methods

Study area

The nuclear power plant at Loviisa is located on the island of Hästholmen (Fig. 1). The cooling water is taken from Hudöfjärden from a depth of 8.5–11.2 m and discharged into Hästholmsfjärden, which is a semi-enclosed basin between the mainland and the islands. The first unit of the power plant came into operation in February 1977, and the second in November 1980. The rated net electrical power of each of the two units was 445 MW in the period 1977–1997 (with full operation) and 448 MW since 1998. The average thermal power discharged from the power plant into the sea was, for instance, 1560 MW in 1994. Water temperature rises by about 10 °C in the condensers of the power plant. The heated water flows into the sea over an embankment, which is 90-m wide and which scatters the effluent over the surface of the receiving waters.

Entrainment of phytoplankton through cooling water systems has been reported to result in markedly reduced productivity and biomasses in cases, when the intake water is chlorinated in order to inhibit biological fouling of cooling water channels (Langford 1990, Huggett and Cook 1991, Poornima *et al*. 2006). However, the effect of this factor is excluded in our study area, because no chlorination or other chemical treatment is used at the Loviisa power plant.

Field and laboratory work

Samples for hydrographical parameters, primary production, and biomass and species composition of phytoplankton were taken at two stations (Fig. 1), one of which was situated in the middle of Hästholmsfjärden, 1 km from the cooling water outlet, and the second in Hudöfjärden (stations 2 and 8, respectively, in Ilus and Keskitalo 1987).

Hydrographical parameters, such as temperature, salinity, pH, transparency, total phosphorus, total nitrogen and oxygen concentration of water, had already been monitored in the study area by Bagge and Niemi (1971) from 1967, and then continued by us from 1971. We took hydrographical samples with a Ruttner or Limnos sampler at the two sampling stations mentioned above, and at eight other stations throughout the area, including the immediate vicinity of the cooling water outlet. Water temperature was determined with the sampler's mercury thermometer, pH with a portable pH meter (Orion Research 401) and oxygen concentration with a modified Winkler method (Koroleff 1979). Salinity, total phosphorus and total nitrogen were analysed at the Finnish Institute of Marine Research in accordance with its standard methods. Salinity was analysed by means of a salinometer and the method is described in Grasshoff *et al*. (1999). A manual method for total phosphorus and total nitrogen analyses was used until 1981 (Koroleff

1979), after which continuous flow analysers and methods given in Koroleff (1983) were used.

Our primary production study (since 1971) follows that of the former study of Bagge and Niemi (1971). Primary production was measured *in situ* with Steemann Nielsen's radiocarbon method (1952) in accordance with the Finnish SFS standard 3049. The sampling was started in April–May and was continued to October–November in 1–2-week intervals in spring and 3–4-week intervals in summer and autumn. Samples were taken in the morning from depths of 0, 1, 2, 3, 5, 7.5 and 10 m (the lowest depth was 12 m in Hudöfjärden from 1971 to 1973). The total depth at the sampling station in Hästholmsfjärden was 11.5 m and 17 m at the sampling station in Hudöfjärden. One ml of $NaH^{14}CO_3$ solution (range 16–104 kBq, lowest activities in the early 1970s) was added to 110 ml of sample water, after which parallel samples were incubated for 24 h in clear and darkened glass bottles. The incubation was finished by adding 0.5 ml of conc. formalin (i.e. 0.2 ml of formaldehyde) to the sample, which was then filtered through a 0.45 μm cellulose-acetate filter. The radioactivity of algae retained on filters was determined with a Geiger-Müller counter from 1971 to 1987 and from then on with a liquid scintillation counter. When the determinations were moved to liquid scintillation counting, the old and new methods were tested parallelly with a large series of samples in 1988. The results obtained with the new method were on an average 6% higher than those obtained with the old method. The results obtained with the Geiger-Müller method and presented in this paper have been corrected to be comparable with those obtained with liquid scintillation counting. The concentration of dissolved inorganic carbon was calculated from pH, temperature and salinity according to Buch (1945). Dark fixation of carbon was subtracted from the light fixation to obtain the final primary production results.

Biomass and species composition of phytoplankton were determined from 1971 to 1982, and in 1985, 1988, 1991 and 1994 in accordance with the preceding studies carried out by Bagge and Niemi (1971) since 1967. Subsamples from depths of 0, 1, 2, 3, 4 and 5 m were combined in one sample. However, from 1971 to 1973, and in 1978 and 1981 the samples were taken from single depths in the water column of 0–10 m (or 0–12 m) (Ilus and Keskitalo 1987). The plankton algae were fixed immediately after sampling with formaldehyde (1971–1974), formaldehyde combined with Lugol's solution (1975–1976) or with Keefe's solution (since 1977). The samples were stored in brown-glass bottles and refrigerated. The samples were studied under a microscope from six months to one year after sampling. The phytoplankton biomass was determined as wet weight by cell counts using Utermöhl's (1958) inverted microscope technique. Each phytoplankton taxon in the sample was identified and counted at a magnification of 500× or 125× using a Leitz Diavert microscope. To ensure that the results were comparable over the years, the biomasses were calculated with the same species-specific cell volumes throughout the study period since 1971 (Ilus and Keskitalo 1987). The nomenclature of the phytoplankton taxa is in accordance with recent taxonomic developments (*see* e.g. Hällfors 2004). *Aphanizomenon* spp. is a filamentous cyanoprocaryote (blue-green alga). It was determined in our former studies as *Aphanizomenon flos-aquae*, but in fact it may have included also other cyanoprocaryote species like e.g. *A. yezoense*.

Statistical calculations

Student's *t*-test was used to study whether Hästholmsfjärden/Hudöfjärden annual primary production value ratios changed significantly after the startup of the power plant. For the corresponding ratios of total phytoplankton biomass, the time-weighted (integrated) means for the growing seasons (May–October, incl. spring maxima occasionally in late April) at a depth of 0–5 m were used in the computations. The ratios of the physical and chemical properties of seawater were calculated from the mean values of the surface water layer.

For the analysis of single phytoplankton taxa, Student's *t*-test with biomass ratios was not appropriate, because zero biomass values occurred and it was not possible to transform the ratio values to a normal distribution. Differences between the growing season means of the sampling stations were calculated for each taxon,

and comparison (before *vs*. after the startup of the power plant) was made with the non-parametric Mann-Whitney *U*-test. The cyanoprocaryote *Aphanizomenon* spp., the dinoflagellate *Peridiniella catenata*, and the diatom *Chaetoceros wighamii* were selected for the analysis, because they were the most abundant taxa in their own systematic groups. Each of them could occasionally contribute to over 50% of the total phytoplankton biomass during the growing season (cf. Results). In addition, *Chaetoceros subtilis* was selected as a species which may favour eutrophied waters (Edler *et al*. 1984).

A stepwise multiple regression analysis was used to discover which environmental factors best explained the changes in primary production and in algal biomasses in the period 1971–1994. The dependent variables were annual primary production (g C m^{-2} a^{-1}) in the entire water column (y_1), total biomass (g m^{-3}) at 0–5 m (y_2), and biomasses (mg m^{-3}) at 0–5 m of *Aphanizomenon* spp. (y_3), *Peridiniella catenata* (y_4), *Chaetoceros wighamii* (y_5) and *C. subtilis* (y_6). The independent variables were: water temperature (°C) (x_1), total phosphorus (mg m^{-3}) (x_2), total nitrogen (mg m^{-3}) (x_3), salinity (‰) (x_4), pH (x_5) and solar radiation (MJ m^{-2}) (x_6). Time-weighted means for the surface layer during the growing season were used for biomass and the independent variables, but sum value was used for solar radiation. The solar radiation values were taken at Helsinki, Kaisaniemi from 1971 to 1987, and at Helsinki-Vantaa airport from 1988 to 1994, as they are the nearest observation stations with continuous data for the study period (Finnish Meteorological Institute, Monthly Reports 1971–1994).

Results

Hydrography

Temperature

The most obvious environmental effect of the power plant has been the increase in temperature of the seawater in the discharge area (Ilus *et al*. 1997). This is true especially in winter, when the warm cooling water has affected ice and temperature conditions at distances of > 10 km west of Hästholmsfjärden (Hari 1982).

In spite of the thermal loading, most of Hästholmsfjärden has been at least temporarily covered by ice in winter. The formation of ice has been assisted by a layer of fresh water which has collected from runoff and river waters from the north-east on the surface of Hästholmsfjärden. The more saline, warm cooling water sinks below the fresh water layer and can spread over wide areas as a thin layer of warm water. The highest temperatures measured under the ice at the depth of 2–3 metres was about 11 °C in Hästholmsfjärden and about 10 °C in Hudöfjärden (March–April 1987). During the winters of the late 1980s and early 1990s, ice conditions were exceptionally poor over the entire coast of Finland. The ice-cover lasted only a few weeks at the Hästholmsfjärden sampling station and 1–2 months at Hudöfjärden.

In open water the thermal effect does not spread far outside Hästholmsfjärden. Near the cooling water outlet the discharged water normally spreads over the surface of the water body, where the heat is dispersed freely into the air. Farther away from the outlet, winds effectively mix the cooling water into the surrounding water masses.

The highest summer temperature (about 25 °C) was recorded at both sampling stations in 1988. The mean surface water temperatures for the whole growing period were also at their highest during this year (Fig. 2). Since the startup of the power plant (1977) the difference between the mean temperatures of the two stations was 1–3 °C (average 2.2 °C), while before startup the natural difference was about 1 °C. The higher mean temperature at Hästholmsfjärden in its natural state was due to its shallowness and semi-enclosed character. On the basis of the above values, thermal discharges increased the mean surface temperature of Hästholmsfjärden by about 1–2 °C from 1977 to 1994.

Salinity

The salinity of the surface water layer in the study area varies from nearly fresh in late winter to 4‰–6‰ in late autumn. In general, the values

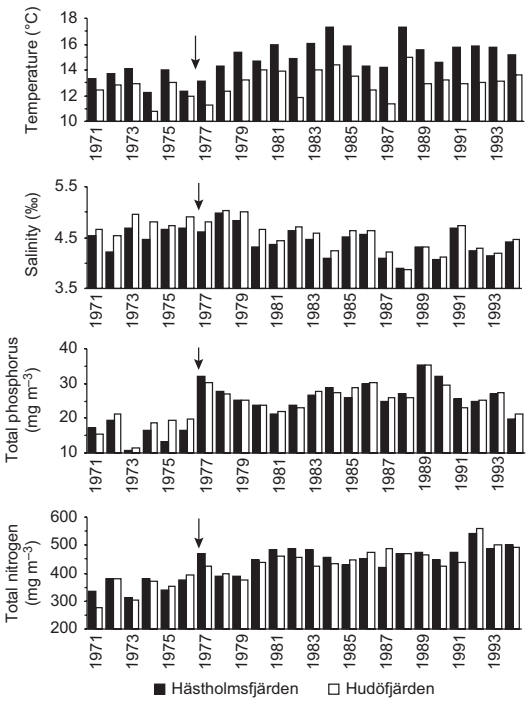

Fig. 2. Time-weighted means of some physical and chemical properties (sampling depth 0 m) in the growing seasons from 1971 to 1994 in the Loviisa archipelago. The arrow indicates the start of the thermal discharges.

have been at their lowest in the distinct fresh water layer under the ice just before the ice melts (minimum records 0.1‰ in Hästholmsfjärden and 0.4‰ in Hudöfjärden). The highest value in the uppermost water layer of Hästholmsfjärden was 6.17‰ in November 1978.

As a whole, the salinity of the water increased in the study area during the 1970s until 1978, after which it decreased towards the end of the 1980s reaching its lowest level in 1988 (Fig. 2). After that there was an increase in the salinity values again. In 1991, the values grew to reach almost the same level as in the 1970s.

The salinity of the water in the study area is mainly regulated by the salinity variations in the entire Gulf of Finland and the Baltic Sea. However, the operation of the power plant has reduced the salinity difference of 0.2‰ between the surface water of Hudöfjärden and Hästholmsfjärden to ≤ 0.1‰ (Fig. 2 and Table 1). This slight change has evidently been caused by the intake of cooling water at a depth of 8.5–11.2 metres at Hudöfjärden.

Nutrients

The concentrations of total phosphorus and total nitrogen varied considerably according to season throughout the study area. In the surface layer, the highest concentrations usually occur in late winter, when plenty of nutrient-rich river waters spread into the area. The highest values measured under the ice at Hästholmsfjärden were 160 mg P m^{-3} (1978) and 2560 mg N m^{-3} (1979).

During the growing period the nutrient concentrations are clearly lower. After the vernal maximum of phytoplankton, the concentrations of total phosphorus generally decrease below 20 mg P m^{-3} and the mean value for the summer months is usually 20–30 mg P m^{-3}. In the autumn, the nutrient concentrations often increase, because the amount of river water rises as a result of autumn rains.

In general, the levels of total phosphorus and total nitrogen were already relatively high in the

Table 1. Ratios between the mean values of Hästholmsfjärden and Hudöfjärden, before and after the Loviisa nuclear power plant came into use (*see* text). Student's *t*-test was used to calculate whether the changes in ratios were significant (two-way analysis).

Parameter	1971–1976	1977–1994	*t*	df	*p*
Temperature	1.08	1.17	4.21	22	< 0.001
Total P	0.90	1.01	2.73	22	0.012
Total N	1.03	1.02	0.21	22	0.834
Salinity	0.95	0.98	3.02	22	0.006
Primary production*	0.86	1.07	3.63	21	0.002
Total phytoplankton biomass*	0.69	1.07	3.01	13	0.010

* 1976 is excluded because the spring maximum was not obtained in Hudöfjärden.

whole study area in the 1960s (Bagge and Niemi 1971), and there was a clear, though variable, increasing trend in their concentrations towards the 1990s (Fig. 2). The increase was primarily equal in the whole area. However, the rise of the mean total phosphorus concentration was somewhat stronger in Hästholmsfjärden than in Hudöfjärden (Table 1).

The total phosphorus concentrations of the surface water increased most in the mid-1970s. After that, the mean concentrations during the growing period stayed at about the same level for a long time, but rose in 1989 to its highest level of about 35 mg P m^{-3}. During the first half of the 1990s the values decreased again (Fig. 2). The total nitrogen concentrations generally rose more evenly throughout the study period. The mean surface water concentrations for the growing period were at their highest of about 540–550 mg N m^{-3} in 1992 (Fig. 2).

Transparency

In general, the transparency of water was relatively low in the study area, being clearly lower in Hästholmsfjärden than in Hudöfjärden. The turbid river waters reduce the thickness of the euphotic layer especially in spring. In 1971–1994, the mean Secchi disc values for the growing periods were 2.8 m at the Hästholmsfjärden station and 3.3 m at Hudöfjärden.

Primary production

The vernal maximum for primary production normally occurs in the Loviisa area at the beginning of the open water period, i.e. 1–2 weeks after the ice melts. At this time the circumstances are very favourable for production: illumination is improved due to the absence of ice and snow cover, nutrients are supplied effectively in surface water due to the inflowing runoff and river waters, and the vernal turnover brings nutrients from deeper water to the surface layer. However, the spring maximum is usually very short and primary production decreases rapidly once the nutrients have been consumed. The maximum may last a few days, but in its natural state up to one third of the annual primary production could be produced in this period (Bagge and Niemi 1971).

In general, the vernal maximum for primary production was higher in Hudöfjärden than in Hästholmsfjärden. This is most probably due to the stronger turbidity of water in Hästholmsfjärden, which limits the thickness of the illuminated layer. During the study period the highest spring maximum value was 2.1 g C m^{-2} d^{-1} in Hudöfjärden (1983) and 1.7 g C m^{-2} d^{-1} in Hästholmsfjärden (1984).

On the other hand, the primary production is usually higher in Hästholmsfjärden in summer. The highest summer value was 0.8 g C m^{-2} d^{-1} in Hästholmsfjärden and 0.7 g C m^{-2} d^{-1} in Hudöfjärden in 1984. The high summer values are often regarded as a sign of eutrophication, but relatively high summer values were measured in Hästholmsfjärden even before the power plant was started up. Thermal discharges have affected the primary production values most significantly in autumn, when the water temperature already started to decrease elsewhere, but was still high in the discharge area. Thus there was often a definite autumn maximum in Hästholmsfjärden.

Since the late 1960s primary production has clearly increased throughout the study area (Fig. 3; *see* also Bagge and Niemi 1971). From 1967–1976 to 1985–1994, the average annual primary production almost doubled in Hästholmsfjärden (58 and 98 g C m^{-2} a^{-1}, respectively), while in Hudöfjärden the corresponding rise was smaller (67 and 95 g C m^{-2} a^{-1}). During the study period the highest values (≥ 130 g C m^{-2} a^{-1}) were recorded in the 1980s (*see* also Table 1).

Biomass and species composition

Phytoplankton biomass showed a regular spring maximum in April–May, after which the values were low in early summer. Biomass tended, however, to increase again in late summer or in autumn (Fig. 4). Considering the study period 1971–1994, the mean phytoplankton biomass of the growing season increased in the Loviisa archipelago from the early 1970s to the late 1980s, but seemed to decrease again in the

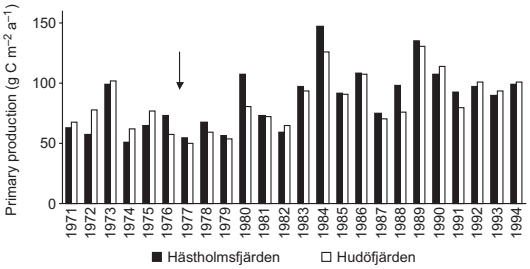

Fig. 3. Annual rates of primary production in the Loviisa archipelago from 1971 to 1994. The arrow indicates the start of the thermal discharges.

early 1990s (Fig. 5). The top mean biomass in 1988 was caused mainly by relatively high summer and autumn values, while the mean of 1991 remained lower in spite of an exceptionally strong spring peak (38.1 and 25.8 g m^{-3} in Hästholmsfjärden and Hudöfjärden, respectively; Fig. 4).

The diatoms and dinoflagellates were the most abundant groups during the spring maximum (Fig. 6). Among the diatoms, *Chaetoceros wighamii* was regularly one of the most abundant species in May both in Hästholmsfjärden and in Hudöfjärden. *Peridiniella catenata* predominated among the dinoflagellates, its proportion being even 84%–85% of the total biomass at both stations in April–May 1975 and over 50% in most springs during the 1970s. Since the early 1980s its biomass remained, however, regularly below 50% of the total phytoplankton biomass in both areas, while the proportion of diatoms became higher (Fig. 6). Typical vernal diatoms, in addition to *Chaetoceros wighamii*, were e.g. *Chaetoceros holsaticus*, *Thalassiosira* spp. incl. *T. baltica*, and *Melosira arctica*.

Several phytoplankton groups occurred together from the middle of summer to autumn. The proportion of cyanoprocaryotes was somewhat higher in Hästholmsfjärden than in Hudöfjärden even before the beginning of the thermal discharges, but this areal difference became clear especially in 1991 and in 1994 (Fig. 6), when this group occurred in July–October in a large percentage in Hästholmsfjärden. In general, cyanoprocaryotes seemed to increase somewhat during the study period 1971–1994, and they predominated in Hästholmsfjärden several months in late summer and in autumn 1994.

Aphanizomenon spp. (Fig. 5) was the most common cyanoprocaryote. However, *Cyanodictyon* spp. predominated in Hästholmsfjärden in August 1991 and *Woronichinia* spp. in September 1994 (58% and 66% of the total phytoplankton biomass, respectively), though they occurred very sparsely in Hudöfjärden during the open-water period. Cryptomonads predominated in Hudöfjärden during the period when *Cyanodictyon* and *Woronichinia* were abundant in Hästholmsfjärden.

At the end of July 1988 the biomass maximum in Hästholmsfjärden (Fig. 4) was caused by the diatom *Chaetoceros subtilis* (Fig. 5) together with *Aphanizomenon* spp. Single specimens of *C. subtilis* were observed for the first time in the Loviisa archipelago in 1976, after which the species has been found frequently in summer and autumn. Its contribution to the total phytoplankton biomass was, however, minute before 1988,

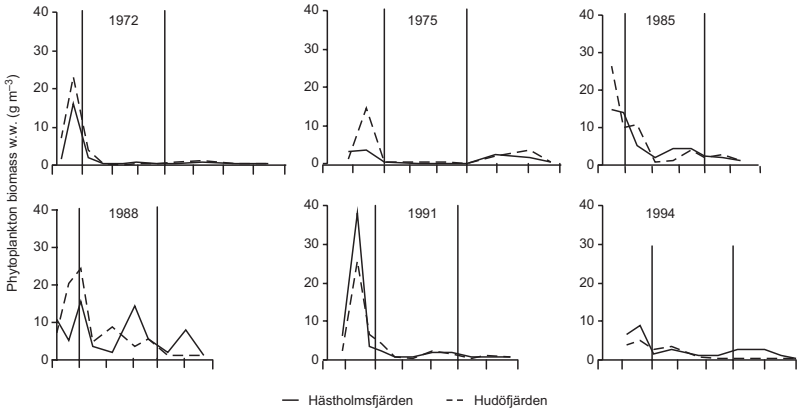

Fig. 4. Phytoplankton biomass (wet weight) at depths of 0–5 m in some years in the Loviisa archipelago. Summer (June–August) is distinguished with vertical lines.

but has been marked since then. (The gaps in Fig. 4 in the 1980s and 1990s are due to the sampling frequency of every three years at that time).

Some diatoms were found frequently in all phases of the growing season, e.g. *Chaetoceros wighamii* and *Skeletonema costatum*. *Thalassiosira pseudonana* was also occasionally fairly abundant in different seasons, being most abundant in the autumn of 1988 in Hästholmsfjärden. Furthermore, the prasinophyte *Pyramimonas* spp. and the chlorophyte *Monoraphidium contortum* were typical, though not predominant species, in all the seasons.

In general, the predominant species were the same through the whole study period 1971–1994. The appearance and occasionally strong increase in *Chaetoceros subtilis* was the most distinguishable exception. On the other hand, the decrease in *Peridiniella catenata* since the early seventies is a marked phenomen, although it has remained one of the most common vernal species even in the eighties and early nineties.

Statistical calculations

The Hästholmfjärden/Hudöfjärden ratios for temperature, total phosphorus concentration and salinity of surface water increased significantly after the power plant started up in 1977, but no change was recorded in the total nitrogen ratio. The ratios for annual primary production and phytoplankton biomass also increased significantly (Table 1).

Considering single taxa, *Aphanizomenon* spp. increased after the startup of the power plant more in the discharge area than in the intake area, and this change was statistically significant (Table 2). The opposite change in the difference was true for *Chaetoceros wighamii* (though with a higher risk level; Table 2). *Peridiniella catenata* was more abundant in Hudöfjärden than in Hästholmsfjärden through the study period 1971–1994. This difference seemed to decrease slightly, but it was not statistically significant. *Chaetoceros subtilis* did not show any significant differences either, although it was particularly abundant in Hästholmsfjärden in the top year 1988.

The regression analysis for the period 1971–1994 (Table 3) shows that temperature best explains the changes in primary production in both areas, and also the changes in total biomass in the discharge area. In addition, temperature best explained the changes in biomass of *Aphanizomenon* spp. in both areas, but not in biomass

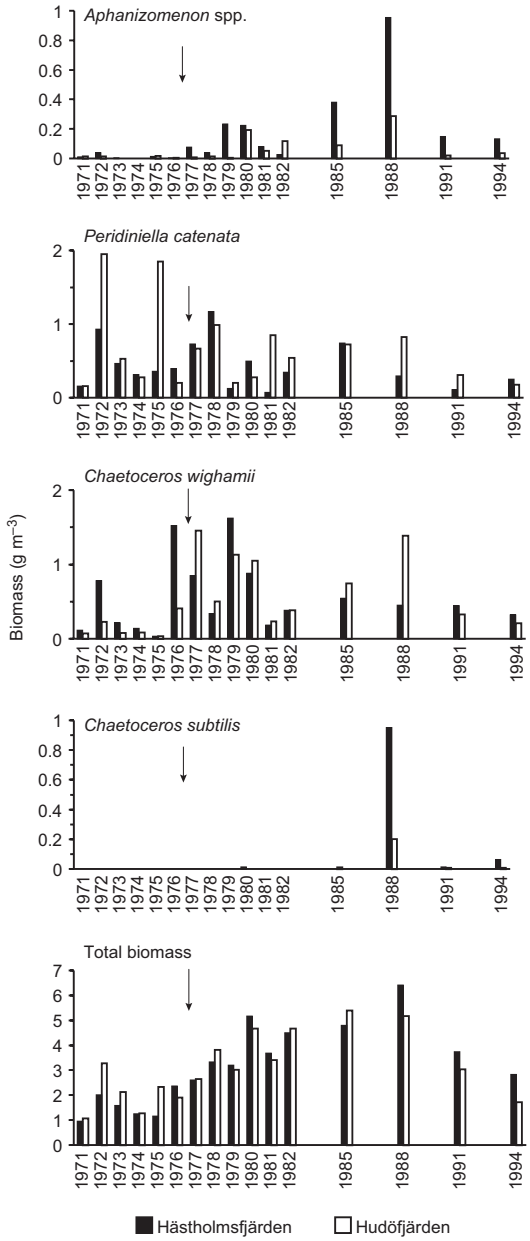

Fig. 5. Time-weighted mean biomasses of some important phytoplankton taxa and corresponding total biomasses of phytoplankton (wet weight; 0–5 m) in growing seasons in the Loviisa archipelago. The arrow indicates the start of the thermal discharges.

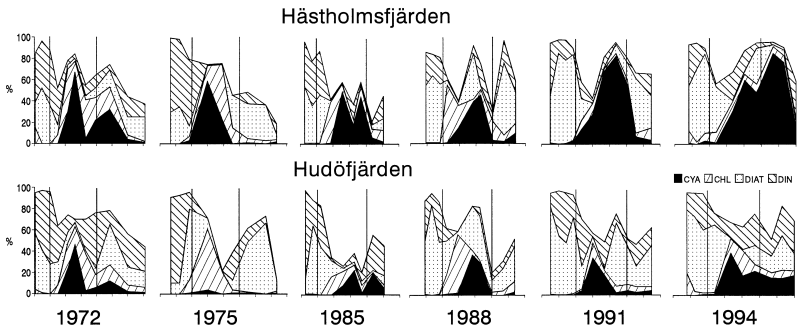

Fig. 6. Percentages of different phytoplankton groups in the total biomass at depths of 0–5 m for various years in the Loviisa archipelago. CYA = Cyanophyta, CHL = Chlorophyta (incl. Prasinophyceae), DIAT = Diatomophyceae, DIN = Dinophyta (excl. Ebriales).

of other taxa. At the intake area, total amounts of phosphorus best explain the changes in total biomass (Table 3).

Discussion

Nutrient concentrations have increased throughout the Baltic Sea since the 1960s (e.g. Cederwall and Elmgren 1990, Pitkänen 1991, Wulff *et al*. 1994, HELCOM 1990, 1996, and 2002). Thus, phytoplankton in the discharge area of the Loviisa power plant has been affected by two main factors: (1) the general increase in the nutrient concentrations, (2) the local increase of temperature. To differentiate between these factors, it was necessary to relate the development of the discharge area (Hästholmsfjärden) to that of the intake area (Hudöfjärden). The areas originally had a somewhat different hydrographic character (Bagge and Niemi 1971). The cooling water flow has equalized some of the differences, and this is an additional factor that must be taken into consideration when interpreting changes in the discharge area.

The cooling water has had an increasing effect on salinity in the discharge area, but only to a less significant extent. A general increase of salinity in the Gulf of Finland resulted in a slight shift of the phytoplankton composition to a more marine character in the late seventies, but this phenomenon was similar in both areas (Ilus and Keskitalo 1987).

The intake of cooling water from a depth of 8.5–11.2 m may be one reason for the greater increase of total phosphorus in Hästholmsfjärden. Another reason may be an internal nutrient load caused by remobilization of phosphorus from sediments in the deep water area of Hästholmsfjärden in anoxic conditions in autumn, associated with the limited exchange of water in this semi-enclosed bay. Nutrient releases from a fish farm may also have contributed to phosphorus increases in the discharge area. The fish farm has been in operation in Hästholmen since 1987; it utilizes the waste heat of the power plant in raising young salmon in the warm outflowing cooling water. In 1988–1992, the average phosphorus load from the fish farm was 0.67 kg d^{-1}, and that of total nitrogen 4.3 kg d^{-1}. The increased ratio for phosphorus in Hästholmsfjärden/Hudöfjärden may have slightly affected the production and biomass of phytoplankton in Hästholmsfjärden, but according to the regression analysis, temperature was a more important factor. In addition, part of the phosphorus increase seems to be a result of the temperature increase and the consequent production rise in surface water, which leads to increased decomposition in near-bottom water (→ increased decomposition → anoxia in the deep area of Hästholmsfjärden

Table 2. Median values of the differences in algae biomass between Hästholmsfjärden and Hudöfjärden (mg m^{-3}), before and after the Loviisa nuclear power plant became operational (*see* text). The Mann-Whitney *U*-test was used to calculate whether the changes in differences were significant (two-way analysis). The year 1976 is excluded, because the spring maximum was not obtained in Hudöfjärden.

Taxon	1971–1975	1977–1994	*p*
Aphanizomenon spp.	1	84	0.014
Peridiniella catenata	–60	–30	0.270
Chaetoceros wighamii	59	–107	0.066
Chaetoceros subtilis	0	0	0.490

→ remobilization of phosphorus from the sediments). The remobilization of phosporus is more efficient than that of nitrogen, and consequently, nitrogen has not increased in the discharge area any more than in the intake area.

Cooling water for the Loviisa power plant is taken from the poorly illuminated part of the productive layer. Therefore phytoplankton biomass might in principle be smaller than if the water was taken from the upper trophogenic layer of Hudöfjärden. However, it is only during the spring maximum that the phytoplankton biomass in Hudöfjärden is greater at depths of 0–5 m than it is in deeper water (Ilus and Keskitalo 1987). Although the depth of the intake may occasionally be a decreasing factor, its significance is evidently small for the biomass of the discharge area, especially because the strong intake flow (50 m^3 s^{-1} in full operation) also catches water from the upper water layers. McMahon and Docherty (1978) showed that in a cold water lake a similar intake-discharge arrangement resulted in the smallest biological and physical changes.

In its natural state, the direction of water currents in Hästholmsfjärden varied irregularly, and the current velocities were low (in the outer straits almost always < 10 cm s^{-1}; Launiainen *et al*. 1982). The operation of the two power plant units resulted in increased outflow through the southern straits of Hästholmfjärden, but the current velocities were still typically less than 10 cm s^{-1} and inflows were even possible (Launiainen *et al*. 1982). As the currents in the discharge area are complicated and the distance from the cooling water outlet to the straits is about 1–2 km, the algae cells evidently remain in the discharge area for a few hours to perhaps several days. Algae have enough time for increased production and biomass in the discharge area, although they also need time to adapt in response to environmental changes (Jorgensen and Steemann Nielsen 1965). Cells which divide rapidly, for example at maximum rate of 2–3 divisions per day (Eppley 1972), can respond to changes in a matter of hours, while cells which divide e.g. once a week, respond very slowly (Harris 1978). We have not

Table 3. Stepwise multiple regression analysis of the environmental data from Hästholmsfjärden and Hudöfjärden in 1971–1994 (1976 excluded, because the spring maximum was not obtained in Hudöfjärden). The forward selection procedure was used ($p \leq 0.10$ to add, $p \geq 0.10$ to remove). *See* the text for the variables. R^2 (%) = coefficient of multiple determination. p_F = significance of the prediction equation (conventional regression equation).

Sampling station	Step	Variable	R^2 (%)	p_F	Prediction equation
Primary production (g C m^{-2} a^{-1})					
Hästholmsfjärden	1	$x_1(t)$	34	0.022	$y_1 = 8.75x_1 - 52.6$
Hudöfjärden	1	$x_1(t)$	25	0.056	$y_1 = 7.30x_1 + 20.1$
Total phytoplankton biomass (g m^{-3})					
Hästholmsfjärden	1	$x_1(t)$	64	< 0.001	$y_2 = 0.976x_1 - 11.07$
	2	$x_2(P_{tot})$	79	< 0.001	$y_2 = 0.795x_1 + 0.115x_2 - 10.91$
Hudöfjärden	1	$x_2(P_{tot})$	40	0.011	$y_2 = 0.178x_2 - 0.75$
	2	$x_1(t)$	59	0.005	$y_2 = 0.551x_2 + 0.164x_1 - 7.52$
Aphanizomenon spp. (mg m^{-3})					
Hästholmsfjärden	1	$x_1(t)$	53	0.002	$y_3 = 135.0x_1 - 1809$
	2	x_4(Salin.)	69	0.001	$y_3 = 109.0x_1 - 387x_4 + 302$
Hudöfjärden	1	$x_1(t)$	41	0.010	$y_3 = 49.5x_1 - 578$
	2	x_4(Salin.)	59	0.005	$y_3 = 22.8x_1 - 157x_4 + 494$
Peridiniella catenata (mg m^{-3})					
Hästholmsfjärden	(No steps)				
Hudöfjärden	(No steps)				
Chaetoceros wighamii (mg m^{-3})					
Hästholmsfjärden	1	$x_2(P_{tot})$	27	0.046	$y_5 = 37.0x_2 - 315$
Hudöfjärden	1	$x_2(P_{tot})$	57	0.001	$y_5 = 77.2x_2 - 1176$
Chaetoceros subtilis (mg m^{-3})					
Hästholmsfjärden	(No steps)				
Hudöfjärden	1	x_4(Salin.)	61	< 0.001	$y_6 = -139x_4 + 6664$

studied division rates in the Loviisa archipelago, but according to the study carried out by Keskitalo (1987a) at Olkiluoto, small-sized flagellates and typical summer taxa tend to have high growth constants (even several divisions in 24 h), while the constants of e.g. *Peridiniella catenata* and some cold-water diatoms are much lower (typically one division in one or more days). The question of how long algae cells remain in the discharge area should, however, be clarified more precisely in future investigations.

In early spring, when the ambient water temperature is ≤ 1 °C, the predominant diatoms tend to increase their production and their cell-specific growth constants strongly due to a temperature increase, as was shown by Keskitalo (1987a) outside the Olkiluoto power plant. Production can be limited by low temperatures (Eloranta 1982). As other factors are not limiting in early spring, the impact of temperature is then important and may advance the start of the phytoplankton spring maximum. The impact of the temperature in early spring is both direct and indirect (improved illumination due to the disappearance of the shading ice-cover). Ilus and Keskitalo (1987) concluded that the shift of the spring phase in the discharge area at Loviisa is 0–2 weeks, depending on the year.

Considering the entire growing season, the most remarkable effect of the warmed water in the discharge area at Loviisa was the increase of *Aphanizomenon* spp., which is the predominant cyanoprocaryote taxon in the study area. As a whole, the proportion of the cyanoprocaryotes in the total biomass increased particularly in the discharge area. The genus *Aphanizomenon* favours warm waters (e.g. Hällfors 2004), which is also true for cyanoprocaryotes in general (Brock 1975). In the discharge area of the Loviisa power plant, the cyanoprocaryotes seemed to displace *Chaetoceros wighamii* to some extent in summer and autumn. *Chaetoceros wighamii* is a species, which thrives well both in cold and warm water (Edler *et al*. 1984). According to our results, the concentration of total phosphorus was the most important variable to explain variations in its biomass. *Chaetoceros subtilis*, a diatom favouring eutrophied water (Edler *et al*. 1984), was very abundant in the discharge area in 1988. However, we did not find evidence that it had been favoured by the discharges in general. The warmed water did not seem to affect the dinoflagellate *Peridiniella catenata*, although it is a cold-water species (Edler *et al*. 1984) and thus in principle might be inhibited by thermal discharges. It was less abundant in the discharge area than in the intake area, but this was true already before the start-up of the power station.

Mean phytoplankton biomass was very high in 1988, but seemed to decrease in 1991 and 1994 in the Loviisa archipelago. However, the spring maxima in 1988 and 1991 were opposite to the mean biomasses (low maximum in 1988, high in 1991). In addition, the decrease in annual primary production was not so clear and the production level was higher still in the early nineties than in the seventies or early eighties. Therefore, it cannot be concluded that the biomass results in the early nineties would indicate a decreasing trend in the Loviisa archipelago. In general, eutrophication has continued in recent decades in the Baltic waters around Finland (Kauppila *et al*. 2004). Kononen and Niemi (1984) stated that many of the dominant phytoplankton species may show marked year-to-year variation which cannot be related to environmental changes.

As a conclusion, temperature has most probably been the main factor in the increase of phytoplankton production and biomass (especially the biomass of *Aphanizomenon* spp.) in the discharge area of the Loviisa power station. Increased nutrient concentrations advanced the phytoplankton production in the Loviisa archipelago in general, but they — as well as other factors discussed above — did not contribute to the relative change between the discharge and the intake area as markedly as temperature.

One question is, to what extent the effects of cooling water can be used to predict changes in biota with a temperature increase, for example, due to possible climatic change. The discharge areas of nuclear power plants can certainly be interpreted as large-scale experimental areas. However, the interactions of temperature with other environmental factors are complex (Nalewajko and Dunstall 1994), and changes in biota are, therefore, difficult to predict. Furthermore, organisms carried by the cooling water

are exposed to rapidly changing temperature and their responses may differ markedly from long-term responses (e.g. gradual colonization, genetic drift). On the other hand, the short generation time of algae is an advantage in the interpretation of phytoplankton responses.

According to the central scenario of the Finnish Research Programme on Climate Change (SILMU) the air temperature will rise in Finland by 2.4 °C from the present until 2050 (Carter *et al*. 1995). In these circumstances the coastal waters will have ice cover about 2 weeks later in the autumn and the ice will have thawed 1.5 weeks earlier in spring than it does now (Leppäranta and Haapala 1996). The FINSKEN project (Developing consistent global change scenarios for Finland) stated that the rise will be about 2–5 °C until 2050 (Jylhä *et al*. 2004), and consequently the effect on coastal waters may be even more pronounced than on the basis of the central scenario of SILMU.

Owing to the low salinity of the Baltic Sea, the biota is poor in species and sensitive to environmental changes (e.g. Kangas *et al*. 1982). Autio *et al*. (1996) considered that the combined effect of temperature and nutrient increases might lead to an enhanced eutrophication of estuaries and coastal waters in the northern Baltic Sea. In the same way as is the case in Loviisa, the species composition of phytoplankton has gradually changed and blue-green algae have become predominant in the discharge area of the Leningrad NPP in the eastern Gulf of Finland (Sazykina 1993).

In conclusion, the algal production may become higher due to both the indirect impact (melting of ice and consequent improved illumination), and direct impact of a temperature rise. In summer, the phytoplankton composition might change occasionally in favour of cyanoprocaryotes and other species which favour warm and/or eutrophied waters, and the frequency of algal blooms might increase.

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